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The Impact of Alien Species on Island Ecosystems: Extended Abstracts of a Symposium, 30 May 1991, Honolulu, Hawaii, XVII Pacific Science Congress[1]

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The unique and fragile ecosystems of small, tropical, oceanic islands are more and more being challenged by both deliberate and inadvertent artificial introductions of alien species. That island species, especially endemics, are highly vulnerable to such introductions, and manifest a rate of extinction far in excess of that of continental species, is becoming widely accepted (e.g., Diamond 1984, Atkinson 1989). The Hawaiian Islands, with their high levels of endemism and spectacular evolutionary radiations in many groups of both plants and animals, arguably have suffered the most severe impacts. The extremely high rate of extinction of the native biota and the modification of native habitat in the Hawaiian Islands are often, but not always, attributable to introduced predators, parasites, competitors, and so forth. It was therefore highly appropriate that this symposium be held in Honolulu.

Many of the presentations dealt with Hawaiian instances, reflecting not only the affiliations of the participants but also the concentration of the problems in Hawaii (and indeed the acute awareness of them, at least on the part of biologists). Nonetheless, the geographical coverage was broad—from Singapore to Hawaii and Guam to New Caledonia and the Society Islands. The subject matter was also broad, ranging widely across the plant and animal kingdoms, from trees to flies, snails to geckos. And, although many participants recounted numerous, depressing case histories, a positive, forward-looking management approach was often intimated and in some instances expounded.

Although much attention focused on the incontrovertible negative impacts of introduced species, controversy was engendered by arguments for the introduction of alien species as necessary agents of biological control in agricultural systems. However, despite strong arguments for biological control as an economic necessity for sustainable agriculture, it is clear that rather rarely are adequate pre-release testing and post-release monitoring of the impacts of new biological control agents carried out (Howarth 1991, Cowie 1992). And once a fragile native animal, plant, or ecosystem has succumbed to the impact of a poorly considered introduction, it is gone forever.

The abstracts are published in the order in which the talks were presented, with the abstracts of two poster presentations (Loope et al. and Medeiros and Loope) at the end. Authors for whom no abstracts are available are listed below:

[1] Manuscript accepted 14 November 1991.

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Hobdy, R. "Lanai—a case study: The loss of biodiversity on a small Hawaiian island." To be published at a later date.
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Two closely related symposia were also held during the congress: "Impact of the Brown Tree Snake on the fauna of Guam," organized by Tom Fritts, and "Introduced aquatic organisms in the Pacific Basin," organized by Jim Parrish and dominated, but not exclusively, by studies on fish. Actively widening awareness of the seriousness of the impacts of introduced species on native, and especially island, ecosystems is an important part of addressing the management of the problem.

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Unwanted Strangers: An Overview of Animals Introduced to Pacific Islands

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ABSTRACT: Since the beginning of traditional island settlement and later Western colonization, animals have been introduced to Pacific Islands by people. Although animals have been transported throughout the Pacific Islands by a multitude of means, human activities have provided one of the best mechanisms. This overview includes terrestrial, freshwater, and marine animals—exclusive of insects—accidentally or intentionally taken to islands. Five periods for such introductions are proposed: (1) During the early periods of settlement and discovery of islands by prehistoric voyagers, traditional life styles were often maintained, and interpreted as "transported landscapes" by several anthropologists. Recently, Roberts (1991) described voyager-related rat dispersal among the islands as early as 3100 to 2500 B.P. (2) The

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exploration period, beginning in the early 1500s with the Spanish and extending until nearly the end of the nineteenth century, saw numerous importations of a wide variety of animals to a broad spectrum of islands. Trans-Pacific galleon trade was among the activities with the greatest impact (Guzman-Rivas 1960). (3) During the late 1800s Western influence and political colonization expanded throughout the Pacific and continued well into the mid-1900s. (4) After World War II and the subsequent political changes, numerous development projects, especially in agriculture, were encouraged and resulted in the introduction of a variety of animals. (5) During the past 20 yr great emphasis has been placed on the development and expansion of terrestrial agriculture and marine and freshwater aquaculture, resulting in a new wave of intentional and accidental introductions.

Not only have larger vertebrates been introduced. Many earthworms are thought to have been taken to the Pacific Islands by Europeans since the 1500s. Fourteen of the 26 species recorded from the southwestern Pacific are known to have been introduced (Easton 1984). Numerous invertebrates were carried on hulls or in bilges of ships (Carlton 1987). Crustaceans and mollusks have escaped from "indestructible" culture ponds following storms or vandalism. Even an onycophoran found in Fiji is thought to have arrived on vegetation originating from Singapore. Several mollusk examples demonstrate disparate impacts on islands. The giant African snail (*Achatina fulica*) was introduced during the 1930s to many Pacific Islands for food and medicinal purposes. Control studies prompted the subsequent introduction of at least three carnivorous snails—one, *Euglandina rosea*, has been the prime cause of extinction of native snails in French Polynesia and Hawaii, at least (see other papers in this symposium). The spread of the predatory planarian *Platydemus manokwari* may be further contributing to similar extinctions (Hopper and Smith 1992). The impact of several species of marine gastropods is unknown. *Trochus niloticus* has been most widely introduced (Bour 1990). Other species include a number of giant clams (*Tridacna* spp.), green snails (*Turbo marmoratus*), pearl shells (*Pinctada* spp.), and green mussels (*Perna viridis*).

A large number of marine and nonmarine fishes have been introduced to many Pacific Islands (Maciolek 1984). More than 60 species are known to have been brought to Hawaii; 36 have become established. Fiji and Guam have received 19 and 16 species, respectively. Several species of "tilapia" have been taken to a number of islands (Nelson and Eldredge 1991) for such purposes as aquaculture, control of aquatic pests, improvement of inland fisheries, and live bait experiments. The marine toad (*Bufo marinus*) began its travels in the 1930s as a biological control effort and is found now on most of the Pacific Islands. The brown tree snake (*Boiga irregularis*) has had a disastrous effect on the fauna of Guam, causing the extinction of most native bird species. Other introduced reptiles include the Caroline anole, green skink, numerous geckos, chameleons, and even a crocodile to Palau. Birds have been widely transported (Lever 1987). A total of 162 species have been released in Hawaii; 45 have become established. New Zealand and Tahiti have seen the introduction of 133 and 56 species, respectively. Among the mammals are a number of classic examples of deleterious introductions—several rat species, house mouse, musk shrew, mongoose, several deer and elk species, and, of course, all the domesticated animals (Lever 1985).

As a generalization, introductions tend to have a negative impact on native floras and faunas. Some marine examples have been commercial successes, but their impacts on native biotas are unknown.

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Management of Alien Species in Natural Areas of Oceania

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ABSTRACT: There is great diversity in the control of alien species in native ecosystems in the Pacific. In some areas (e.g., Hawaii and New Zealand), extensive programs are conducted because of the abundance and impact of these species on native ecosystems, extensive scientific and fiscal resources, and a strong conservation lobby (see, e.g., Stone and Scott 1985, Towns et al. 1990). In other areas (e.g., Samoa), alien species are minor components of the flora and fauna and are generally not an important threat to native ecosystems. In most island systems, they are tolerated because resources to control them are inadequate, the alien is serving a needed function (e.g., erosion control), or conservation of native ecosystems is not a high priority. The biggest problems are on the larger island systems where agriculture has been or still is an important element of the economy. Large numbers of species are introduced, deliberately or inadvertently, many of them escape, and some become noxious.

Mechanical control techniques are frequently used, but one drawback is the disturbance caused by the management action. Large mammals (e.g., feral cattle, goats, pigs, sheep) are relatively easily controlled using exclusionary fencing, hunting, and trapping in various combinations. The cost is high but the benefits substantial. Smaller mammals (e.g., cats, mongooses) are managed using baited traps in target areas where reinvasion cannot be controlled. Some plants are easily uprooted. Others that do not resprout are readily controlled by felling (e.g., conifers), but most need further chemical treatment. Chemical methods are used extensively. Small mammals have been successfully controlled or eradicated from small islands, principally in New Zealand, using poisons. Nontarget impacts must be minimal or nonexistent. Herbicides are most effective where research has determined the lowest effective dose to protect other species and minimize residues in the habitat. Biological control is an increasingly important management option though there are some legitimate concerns. Unfortunately, biocontrol agents continue to be used even when their negative impacts are known (e.g., the recent introduction of *Euglandina rosea* to Ta'u). Agricultural interests have dominated the use of this technology in the past, but an interagency program against forest pests has

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been established in Hawaii and is working effectively. A number of agents have been released against *Myrica faya*, *Passiflora mollissima*, and *Ulex europea*. Exploratory research is being conducted on *Rubus argutus* and *Psidium cattleianum*.

The problems facing a manager often seem overwhelming. Some of them seem to be impossible to overcome (e.g., controlling or eradicating alien passerine birds). A recent positive approach focuses management efforts on small critical areas. The lessons learned and the psychological boost from success allows increase of the area managed or addressing of other high-priority areas. The remarkable successes in New Zealand and Hawaii have demonstrated a number of points: (1) management resources must not be overextended; (2) once a program is started it must be seen through to its conclusion; (3) careful planning and budgeting are essential; (4) positive goal statements are important; (5) education, through interpretation programs, provides crucial support; and (6) many capable volunteers are available. The bottom line is, "where there is a will there is a way."

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Biological Control in Island Ecosystems: Cornerstone of Sustainable Agriculture or Threat to Biological Diversity

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ABSTRACT: Several papers in recent years have focused increased attention on the detrimental effect that classical biological control introductions may have on nontarget flora and fauna (Howarth 1991). Although laudable for bringing to light a problem previously ignored, some of these concerns have been overstated to the point where they are hindering the most careful use of predators and parasitoids for effective and nonpolluting pest control.

Within the State of Hawaii there are abundant safeguards in place to ensure the safety of biological control introductions. Knowledge of candidate insect taxonomy and host relationships strictly narrows the field of potential natural enemies. Federal and State permits must be obtained from the Department of Agriculture before any shipments of natural enemies can be imported. Federally licensed quarantine facilities are used to receive and confine candidate species while additional testing is done on host/prey affinities, and pathogens or hyperparasites are removed. Finally the Board of Agriculture (and two professional advisory committees) review relevant data and must give approval before any organisms can be released in the field.

Hawaii has had more insects introduced for biological control than any other region in the world, yet has an outstanding safety record to document the efficacy of existing safeguards. Of 639 arthropod species introduced, very few have even been implicated in important nontarget effects, and these implications have been based on inaccurate information, speculation, and guilt by association (see discussions in Zimmerman 1978, Gagné and Howarth 1985; rebuttals in Funasaki et al. 1988, Lai 1988).

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Howarth (1985) has gone so far as to call for a comprehensive environmental impact statement (EIS) before any biocontrol organisms can be introduced to Hawaii. Preparation of an EIS is a long, involved process that often takes years of work and millions of dollars to accomplish. There is no doubt that many successful biological control projects would never have been undertaken in the first place if faced with the daunting economic and bureaucratic hurdles involved in preparing an EIS. The result would have been increased pesticide use and possible toxic chemical threats to the very arthropod species that biocontrol critics are trying to conserve.

In Hawaii, as in all island ecosystems where people live and grow food, crops must be protected to some degree from insect pests. Any pest control measure entails a finite degree of environmental risk. In the hundred years since its widespread application, biological control has been the most environmentally safe and cost-effective pest management tool available. Before clamping down too tightly and seriously interfering with the practice of biological control, critics should realistically consider the alternatives.

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Environmental Impacts of Species Purposefully Introduced for Biological Control of Pests

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ABSTRACT: The intentional introduction of alien species has been an important strategy for controlling pests. However, assuring the public that the method is safe begets the philosophy that alien species invasions are acceptable and even encourages additional damaging introductions by well-intentioned persons. All pest control methods entail risks; biocontrol is nearly irreversible and involves special risks because the agents are self-perpetuating and self-dispersing. Accurate information on potential impacts of all control methods needs to be considered to choose the least risky technology for each pest problem (Miller 1990, Howarth 1991, Howarth and Ramsay 1991).

Because biocontrol has generally been considered safe, the environmental effects of purposely introduced organisms were rarely looked for or recorded. Most of the evidence of damage has been gathered serendipitously; thus many severe impacts certainly occurred without being documented. Biocontrol importations have failed to control the pest, enhanced pest populations, synergistically interacted with other organisms to enhance pest

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problems, affected public health, expanded their host ranges to feed on or compete with valuable nontarget species, and caused species extinctions (Howarth 1991). In no other aspect is the biocontrol strategy more in conflict with other human values than as a cause of species extinctions. Biocontrol agents have been strongly implicated in the extinctions of nearly 100 species worldwide, and many more species are imperiled. Some of the best-documented extinctions involve biocontrol introductions (Murray et al. 1988, Howarth 1991, Morris 1991).

Biocontrol agents can feed on all available suitable hosts, disperse to occupy all suitable habitats, and affect, either directly or indirectly, associated species, including nontarget species. The outcome of these encounters may range from negligible impacts to extinctions, but small changes often lead to large ecological effects. Examples of significant problems are found in all regions. The relative level of risk can be correlated with the permanency of the agent in the environment, its host range, habitat range, genetic plasticity, behavior, mutualistic relationships, and vulnerability of the target region (Howarth 1991). Vertebrates, diseases, and certain eurytopic and polyphagous species have been most damaging.

Better protocols are urgently needed to review and regulate all purposeful introductions and to improve quarantines. Biocontrol requires good systematics studies, including preservation of well-curated voucher specimens. The importing agency should support long-term postrelease studies that monitor both the efficacy of the agent and its effects on nontarget species. Data from these studies are needed to make biocontrol a predictive science. Until the outcome of intentional importations can be predicted, biocontrol will remain part of the problem of alien species invasions rather than a solution.

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Ecological Impact of Alien Plant Species in Singapore

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ABSTRACT: The Republic of Singapore consists of 58 islands with a total area of 626.4 km^2. The main island is separated from Peninsular Malaysia by the Straits of Johore, which are only 600 m across at their narrowest point. Some 2100 seed plants are native to Singapore (Turner et al. 1990). An unknown, but probably considerable, proportion of these have become extinct over the last 150 yr very largely because of human clearance of the natural vegetation, which was closed forest, 82% by area lowland tropical rainforest, 13% mangroves, and 5% freshwater swamp forest (Corlett 1991). Human activity has also led to the introduction of many alien plant species (Corlett 1988), of which ca. 200 have become naturalized. In general, these species appear to have had little or no effect on the few remaining areas of primary vegetation such as

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lowland rainforest or mangroves. For instance, the only common exotic in the ca. 50 ha of primary lowland dipterocarp forest in Bukit Timah Nature Reserve is *Clidemia hirta*. Alien species may hasten the decline of very small remnant patches, as witnessed by the "Gardens' Jungle" of the Singapore Botanic Gardens where the climbers *Dioscorea sansibarensis* and *Thunbergia grandiflora* are smothering regeneration of the 4-ha forest (Whitmore in press). *Adinandra dumosa* is the dominant species in a secondary forest facies found on highly degraded soils produced by unsustainable agriculture (Holttum 1954). This community also seems to be able to prevent invasion by exotics. But on apparently more nutrient-rich sites successional communities dominated by aliens such as *Mimosa pigra*, *Acacia auriculiformis*, *Paraserianthes falcataria*, and *Spathodea campanulata* may develop (Corlett 1988). *Eichhornia crassipes* and *Salvinia molesta* became problems on several reservoirs during the 1970s and 1980s (Wee and Corlett 1986). Control measures have now succeeded in keeping the water clear of these plants, though at considerable expense.

The native flora of Singapore possesses few species well suited to disturbed conditions. It is not surprising therefore that contemporary Singapore's urban landscape is dominated by exotic species. Even quite small patches of native vegetation, especially ones of relatively harsh environments such as those inundated by salt water or with extremely acid, infertile soils, appear resistant to invasion unless they are disturbed. Disturbance clearly favors pantropical weed species.

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Miconia calvescens: A Threat to Native Forests of the Hawaiian Islands

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ABSTRACT: Managers of protected areas on oceanic islands usually recognize the need to eliminate aggressive alien species at an early stage before these species become fully established and reach the boundaries of the protected area. The principle is deceptively simple. In practice, however, there are often severe obstacles to success. Few records of successes or failures are reported in the conservation biology literature. We report here on our initial experience on Maui with *Miconia calvescens*, a tree native to mid-elevation forests of Central and South America, with large, dark green leaves with maroon undersides.

Because of the perceived attractiveness of *Miconia calvescens* as an ornamental, it was introduced in the 1970s through the horticultural industry to at least three Hawaiian islands: Hawai'i. O'ahu, and Maui. Whereas it is said to show little evidence of invasive

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tendencies on O'ahu, it is spreading aggressively in several areas in the vicinity of Hilo on Hawai'i Island and near Hāna on East Maui. The situation on Maui is such that control appears feasible if prompt and concerted action is taken. Tiny, bird-dispersed seeds are produced after about 4–5 yr of vegetative growth; each tree has the capacity to produce millions of seeds annually. Unlike most species, it has the ability to establish seedlings in either open sunlight or dense shade.

Miconia calvescens is known to be a species with great invasive potential; it has, in the past two decades, extensively invaded forests of Tahiti and Moorea, apparently even those at high elevations (above 1200 m). Haleakala National Park learned about the threat of *M. calvescens* when the senior author (who had been in Tahiti in 1977 and 1988, and was thus attuned to the threat from this plant) noticed a single tree growing in Ali'i Gardens near Hāna. After an inquiry to the owner of Ali'i Gardens in January 1991, our knowledge of its status on Maui advanced rapidly. Our approach at Haleakala National Park has been to attempt, at least on a small scale, to publicize the problem and to raise community consciousness and concern. We distributed "wanted" posters illustrating *M. calvescens* around windward East Maui and now feel that we have a fairly good knowledge of the local distribution of this plant, although there are undoubtedly populations of which we are unaware. As of mid-1991, we know of seven populations. *Miconia calvescens* probably first arrived on Maui with a horticultural shipment to Helani Gardens, near Hāna, in the late 1970s. Founding individuals have grown to nearly 0.3 m in diam. and over 10 m tall and produced abundant seedlings locally. *Miconia calvescens* appears to merit special concern on East Maui since few other plant species are invasive in forest situations above 1200 m elevation. *Miconia calvescens* grows at 340–1800 m in its native habitat from southern Mexico to Bolivia and Brazil.

It is becoming increasingly obvious that there is no effective mechanism operating in Hawai'i to keep aggressive alien species such as *M. calvescens* from being brought in. Haleakala National Park will be exploring more fully the feasibility of a cooperative effort with nurseries in the Hāna area, with the state Division of Forestry and Wildlife, with the East Maui Irrigation Company, with The Nature Conservancy, with local conservation groups, with Hana Ranch, and with other private landowners, to eradicate *M. calvescens* from Maui. All these groups have been highly cooperative so far. A major preliminary effort toward *M. calvescens* eradication on Maui was undertaken in Helani Gardens during four days in June–July 1991 by Haleakala National Park staff and volunteers. Helani Gardens owner Howard Cooper gave full support to the effort, and Keola Hana Maui gave permission to remove *M. calvescens* plants on their land adjacent to Helani. A total of 9320 *M. calvescens* plants were removed, with 97% less than 5 cm in diam.; 24 plants were larger than 20 cm in diam. We estimate tentatively that this effort removed 90% or more of the *M. calvescens* plants on Maui. Follow-up is obviously necessary and is planned.

Ecophysiological Factors Influencing *Myrica faya* Invasion of Hawaii Volcanoes National Park

CINDY C. LIPP[9]

ABSTRACT: The establishment of the alien tree *Myrica faya* in the near-native ecosystems of the Hawaiian Islands seriously threatens the integrity and survival of these ecosystems and the endemic plant populations and communities within them (Smith 1989). The open-canopied forests of the seasonal submontane

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zone of Hawaii Volcanoes National Park (HAVO), on the flank of Kilauea, have been aggressively invaded by *Myrica faya* over the past 20 yr and the ecosystem-level consequences of this invasion have been reported by Vitousek and Walker (1989). Community-level vegetation structure and habitat characteristics can either facilitate or hinder the invasion of alien plants depending on individual species requirements for establishment. Potential habitat and species-specific physiological limitations to the invasion of certain HAVO sites by *Myrica faya* were examined in this study.

The closed canopied rainforest is dominated by a *Metrosideros polymorpha* overstory with either a *Cibotium* sp. or *Dicranopteris* sp. understory. The rate of invasion of *M. faya* into rainforest has been much slower than in the seasonally dry zone. The canopy structure of these community types appears to play a significant role in *M. faya* establishment. Though germination occurs, the survival of *M. faya* germinants under low light conditions is relatively low. Light availability is a strong factor limiting *M. faya* growth (Vitousek and Walker 1989). The leaf area index (LAI) of *Metrosideros/Cibotium* forest can range from 2 to 3 m^2/m^2, which permits enough light to get through the canopy for the slow but persistent recruitment of *M. faya*. Established *M. faya* trees join the mid- and upper-canopy levels, increasing the LAI to about 5 m^2/m^2, which reduces understory light levels to an extent such that further recruitment of its own species as well as others cannot occur. *Metrosideros/Dicranopteris* forest sites have an LAI of 2 m^2/m^2 above the fern understory and LAI of 6 m^2/m^2 below the fern canopy. Survival and growth of young *M. faya* seedlings occurs only where the *Dicranopteris* canopy breaks down. Low seed input, low levels of seed germination, and low light availability for growth are factors restricting the invasion of *M. faya* into rainforest habitat.

Myrica faya, a tree adapted to moist mid-elevations in its native Canary Islands (D. Mueller-Dombois, pers. comm.), survives in the seasonally dry habitats of HAVO through two mechanisms: a seasonal adjustment of osmotic potential that permits some maintenance of leaf turgor as soil water becomes less available, and stomatal control of water loss. If these water loss control mechanisms are unable to maintain leaf turgor during a period of drought, leaves are dropped. Resprouting and development of new leaves have been observed when soil moisture levels are high again.

The seasonally dry zone consists of shrub and grasslands dominated by the alien grasses *Melinus minutiflora* and *Schizachrium condensatum* (Hughes et al. 1991). High light conditions in this zone favor *M. faya* growth and establishment though seed inputs are relatively low. Seed germination trials revealed that the amount of *Melinus minutiflora* canopy cover influences *M. faya* germination and survival. Without plant cover, only 21.6% of viable *M. faya* seeds tested germinated, with only 4% of germinants surviving to 4 months of age, while under a full *Melinus minutiflora* canopy there was 84% germination of viable seeds, with 28% survival. *Melinus minutiflora* cover appeared to ameliorate the effect of periodic conditions of low moisture and high soil temperatures that would be experienced by exposed seed. There is evidence that when soil water availability is low some degree of competition for soil water occurs between *Melinus minutiflora* and *M. faya*. One of the consequences of the invasion of the alien grass *Melinus minutiflora* into the seasonally dry ecosystems of HAVO appears to be the facilitation of the invasion of another alien species, *M. faya*.

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Impact of Mealybugs on an Endangered Tree, *Serianthes nelsonii*, in Guam and Rota

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ABSTRACT: *Serianthes nelsonii* Merill is one of two species of plants of Guam listed in the U.S. Federal list of endangered plant species. Currently the world population consists of 60–90 trees on Rota and two trees in the limestone forest on Guam. Seedlings, but no saplings or young trees, are found on both Guam and Rota.

Growing tips of these trees are often distorted by mealybugs of one or more of three species. Of these, *Dysmicoccus neobrevipes* and *D. brevipes* are neotropical species that were introduced to Guam within the last 300 yr. *Planococcus citri* is a cosmopolitan species but was not collected on Guam until 1958 (Beardsley 1966). Two trees on Guam were monitored beginning September 1990. On every sample date more than half of the branches on one suburban tree had mealybugs on them, and infested tips generally died as a result of mealybug damage. By December 1990, only about a quarter of the tree had green branches. At that time a typhoon defoliated the tree, and it subsequently died. The combination of severe mealybug damage and defoliation apparently killed this tree. The trees in the forests on Guam and Rota had fewer mealybugs. On the monitored forest tree on Guam, mealybugs persisted only in parts of the tree patrolled by the ant *Anoplolepis longipes*. Control of the ants resulted in disappearance of the mealybug colonies.

Seeds collected on Rota were planted in pots on Guam in October 1990. The seedlings were continually infested and reinfested by *D. brevipes*, which established below the soil line and killed the seedlings if not controlled. Mealybug infestation of the roots appears to have killed most of the trees reared in nurseries and some wild ones. Both *P. citri* and *D. neobrevipes* are found attacking roots of other plants, although they have not yet been observed on roots of *S. nelsonii*. *Dysmicoccus brevipes* also caused distortion of *S. nelsonii* leaves. On seedlings, ants move the mealybugs from the branch tips to the roots. One uncaged wild seedling on Guam was fatally browsed by a large mammal, probably a deer, during the months we were monitoring the trees. Deer are exotic to Guam, and readily consume seedling trees. The relative role of mealybugs and deer in preventing *S. nelsonii* seedling survival to the sapling stage is not known, though it could be assessed on Rota. All herbivore species involved seem to be exotic to the island. *Serianthes nelsonii* is apparently not adapted to this suite of herbivores, and the result is a gradual decline of the populations because of a lack of successful recruitment.

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Impact of Intentionally and Accidentally Introduced Biological Control Agents on Unintended Hosts, *Hypolimnas anomala* and *H. bolina* (Lepidoptera: Nymphalidae), on Guam

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ABSTRACT: Guam has had 20 species of butterflies and skippers recorded. Most are widely distributed Asian species, but two endemic species and one subspecies are present. Since 1945, five species of butterflies have become extremely rare or extinct. These include the endemic butterflies *Euploea eleutho* (Latreille & Godart) (Danaidae) and *Vagrans egistina* (Latreille & Godart) (Nymphalidae) and the widespread species *Appias paulina* Cramer and *Papilio xuthus* (L.) (Papilionidae). *Hypolimnas octocula marianensis* Fruhstorfer is rare, and possibly endangered.

Until recently, biological control has been considered a safe method of pest control with few negative impacts. However, Howarth (1991) has pointed out some environmental problems caused by biocontrol introductions, including species extinctions. On Guam, over 100 species of exotic organisms have been deliberately introduced for biological control (Nafus and Schreiner 1989). Of these, 27 species were released against seven lepidopterous pests including one butterfly and one skipper. To see if movements of biological control agents to nontarget butterflies was a possible problem, from 1987 to 1989 I monitored mortality factors affecting juvenile stages of the nymphalid butterflies *Hypolimnas anomala* (Wallace) and *H. bolina* (L.). Both species were attacked by the same species of native, accidentally introduced, or deliberately introduced parasitoids and predators. Parasitoids killed 2.4% of the eggs of *H. anomala*, and ants removed about 25%. *Hypolimnas bolina* had 40% of its eggs killed by parasitoids and 35% eaten by ants. One species introduced for biological control, *Trichogramma chilonis* Ishii, was found parasitizing the eggs of both species, but it caused little mortality. Most parasitization was by the native species *Telenomus* sp. and *Ooencyrtus* sp. Most causes of larval mortality could not be quantified. No larval parasites were reared for either species. *Hypolimnas anomala* suffered heavy mortality from a disease, probably a virus. Both species were attacked by ants and by two exotic self-introduced predators, *Hierodula patellifera* (Serville) (Mantidae) and *Eocanthecona furcellata* (Wolff) (Pentatomidae). *Hierodula patellifera* and *E. furcellata* killed less than 3% of larvae. The toad *Bufo marinus* L., which was introduced as a biological control agent in 1937, ate a few larvae of *H. anomala*. The major pupal parasite was *Brachymeria lasus* (Walker), a deliberate biological control introduction. *Brachymeria lasus* parasitized 19% of the pupae of *H. bolina* and 2.9% of *H. anomala*. Ants attacked a minimum of 17% of *H. anomala* pupae and 7% of *H. bolina*.

Two deliberately introduced biocontrol agents attacked the nontarget butterflies. One, *Trichogramma chilonis*, was uncommon. The other, *B. lasus*, was present only during periods of the year when *H. bolina* was abundant. Neither biocontrol agent played a role in the possible extinctions of *A. paulina*, *P. xuthus*, or *E. eleutho*, as these species had disappeared before biocontrol agents were introduced. *Vagrans egistina* could have been affected by *B. lasus*. *Vagrans egistina* is still present on Rota, where it has coexisted with *T. chilonis* since the late 1930s.

Ants preyed on juvenile stages of both butterflies, and overall were the most important mortality factor. Ants killed all *H. bolina* eggs and larvae during certain parts of the year. All the ants were self-introduced tramp species, some of which were recent introductions. The role of exotic ants in the extinction of island species cannot be overemphasized. Ant predation takes place quickly, leaving little evidence, and can be easily overlooked. Without detailed life history data, the role

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of ants, deliberately introduced biocontrol agents, or accidentally introduced generalists (Funasaki et al. 1988) cannot be disentangled, making it difficult to ascertain causes of extinction and develop appropriate preventive measures.

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Alien Predators and Decimation of Endemic Hawaiian Tree Snails

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ABSTRACT: The disappearance of more than half of the 1000+ endemic Hawaiian terrestrial snail species since the mid-1800s is recognized, and its causes are relatively well delineated: destruction of forest habitats, habitat degradation from the introduction and spread of alien plant species, and removal of snails by shell collectors and introduced predators. However, most knowledge of Hawaiian gastropod extinctions is based on incomplete surveys, anecdotal observations, and post facto deductions. Recently, we have documented destruction of endemic tree snail populations by two different alien predators. In early 1988 a well-studied population of the endemic tree snail *Achatinella mustelina* was invaded by a large number of rats that ate most of the resident snails. Because the individual snails in the population, which numbered about 200 at the time of the disaster, were marked with identifying codes and measured, it was possible to determine with great accuracy the demographic consequences of rat predation upon the snails. The rat invasion was not allowed to run a natural course (if such could be possible with an alien species), but instead was mostly destroyed with poison bait. The rats ate most of the largest snails, including the mature adults and those that would have become mature during the next 2 yr. In addition, a typical annual summer die-off of very small juveniles occurred soon after the rat invasion. After six months the snail population had been reduced by more than two-thirds, and it consisted almost entirely of mid-sized snails 1–3 yr of age. In the $2\frac{1}{2}$ yr since the invasion and subsequent destruction of the rats, new adult snails have migrated into the population, some survivors have grown to maturity, the birth rate is increasing, and the population is growing.

At a second site, a marked population of *A. mustelina* is being destroyed by the alien predatory snail *Euglandina rosea*. This predator eats snails of all sizes, no remedies have been discovered for its removal, and almost all of the tree snails have been eaten during the last 2 yr. We thus conclude that populations of native tree snails, even though characterized by very slow growth rates, late ages at maturity, and low fecundities, can recover fairly rapidly from predator invasions if the predators are removed rapidly enough. Unfortunately, in most cases the predators cannot be selectively destroyed, and their depredations continue for years, eventually driving the native snails to extinction. The latter scenario has occurred repeatedly where the predatory snail *Euglandina rosea* has been introduced to Pacific islands in well-intentioned attempts to control the giant African snail, *Achatina fulica*.

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Introduced Land Snails in New Caledonia: A Limited Impact in the Past, a Potential Disaster in the Future

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ABSTRACT: At least 22 species of land snails have been introduced to New Caledonia (Solem 1964, Tillier 1982 and unpublished data). Most of them are "tropical tramps," probably introduced within the last 100 yr, and do not seem to have had any direct impact on natural environments. But four—*Achatina fulica* Bowdich, *Euglandina rosea* Férussac, *Gonaxis quadrilateralis* (Preston), and *G. kibweziensis* (Smith)—have been introduced deliberately within the last 20 yr. *Euglandina rosea* and *Gonaxis* spp. are considered potential predators on juveniles and eggs, respectively, of *A. fulica* and, therefore, potentially useful in control of *A. fulica*. However, because these predators are not species-specific and endemic snail species are generally smaller and more vulnerable than *Achatina*, predation by these introduced species may lead to the extinction of most, if not all, of New Caledonia's extraordinary endemic land snail fauna of several hundred species (Solem 1961, Tillier and Clarke 1983).

In New Caledonia, introduced species of land snails are as yet absent from environments considered by botanists as untouched by Man, whereas autochthonous species are absent from environments considered totally secondary. Analysis of the malacofaunal composition at 284 stations where collections including endemic species have been made shows that introduced species occur in 78% of lowland low-rainfall stations (altitude <250 m; rainfall <1500 mm), in 37% of lowland high-rainfall stations, in 21% of medium-altitude stations (altitude 250–750 m; rainfall >1500 mm), and in no station above 750 m altitude (all very high rainfall). Considering only stations with at least one endemic species, in lowland highly secondary stations with low rainfall up to seven introduced species may occur sympatrically, whereas in other stations no more than three introduced species are found. This analysis leads to the strong suggestion that, at least in the case of the "classical" tropical tramps, introduced species are an index of the secondary nature of the environment, but probably are not themselves a danger to primary environments, which they do not seem to invade until the environments are modified. This optimistic conclusion is good news for conservation of native species where totally primary environments are preserved, but we still do not know what interactions, such as competition, might occur between native and introduced species as secondary environments spread and become more common. Carnivorous snails may behave differently.

Achatina fulica is about the same size as the endemic litter-dwelling *Placostylus* spp. and is a good candidate for competition with them in lowland, moderately secondary environments that have been reached by the former and where the latter still occur. *Achatina fulica*, therefore, is a potential additional threat to the remaining populations of *Placostylus* spp., which are relicts of species once widespread over all the territory. It is too soon to determine whether *A. fulica* will spread more widely than other previously introduced species. There is no good record of the spread of *Gonaxis* spp. but this may be due to their low densities and dispersal rates. *Euglandina rosea*, introduced 10 yr ago to control *A. fulica*—without effect—is spreading south of Nouméa at a speed that, although not precisely estimated, is probably similar to that in Polynesia (about 1 km per year). Although it has not reached any primary environment yet, it will probably do so within a few years and is a real threat to several hundred endemic land snail species.

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Introduced Ampullariid and Viviparid Snails in Hawaii

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ABSTRACT: At least five species of the closely related freshwater prosobranch families Ampullariidae (so-called "apple snails") and Viviparidae have been introduced to the Hawaiian Islands and become established. They belong to three genera: *Cipangopaludina* (Viviparidae) and *Pila* (Ampullariidae), which are Asian, and *Pomacea* (Ampullariidae), which is found in tropical and subtropical regions of the Americas and the Caribbean. Their distributions in Hawaii have been ascertained from material in the Bishop Museum, much of which was collected during 1990–1991, although collections of viviparids go back to 1926 and of ampullariids to 1962. The latter date probably represents the earliest record of Ampullariidae in the Hawaiian Islands, but both Caum (1928) and Clench and Fuller (1965) recorded Viviparidae. The viviparid *Cipangopaludina chinensis* has been found in the wild only on the islands of Maui (three localities) and Hawaii (two localities). Ampullariidae are more abundant and widespread. *Pomacea canaliculata* is the most widespread, being found in the wild at five locations on Oahu, one on Maui, and one on Kauai, often in large numbers. *Pomacea bridgesi* has been found on Oahu, Maui, and Hawaii. *Pomacea paludosa* is known only from a single locality on Maui. *Pila conica* has been found at two localities on Maui and one on Oahu. Frequently, more than one species co-occur, notably at Keanae, Maui, where all except *Pomacea bridgesi* have been found.

All species have been deliberately introduced for one or both of two reasons: (1) for harvesting, or even culture, for food, and (2) as domestic aquarium snails. Most introductions probably took place during the 1980s; new introductions are almost certainly continuing. Aquaculture projects on Oahu, Maui, and Kauai have involved *Pomacea canaliculata*, *P. bridgesi*, and *Pila conica*. But, although some people see the culture of these snails in positive terms as a source of revenue, others complain that their taro crop (the traditional staple of the Hawaiian Islands) is being seriously damaged by increasing snail populations. The snail farmers insist that their snails cannot escape and do not pose a problem, but experience suggests that, eventually, escapes will occur. In Southeast Asia, *Pomacea* spp., introduced deliberately for culture as a source of food, are now serious pests of rice and other crops (Cheng 1989, Mochida 1988, 1991).

Although the snails are at present predominantly, but not exclusively, found in taro-growing areas, little concern has been expressed over other more fundamental ecological impacts (e.g., destruction of native freshwater habitat and competition with native freshwater snails) that might be caused should these introduced snails expand into Hawaii's unique and fragile native ecosystems

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(cf. papers in Stone and Stone 1989). Their potential for disease transmission (Keawjam 1986) is probably important but has barely been considered.

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Restoration of Soil Microarthropod Populations after Feral Pig Removal in a Hawaiian Rainforest Ecosystem

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ABSTRACT: Soil microarthropods (0.3 to 5 mm in size) are an abundant, cosmopolitan group with high species and life form diversity. Their numbers, biomass, species compositions, and distributions depend on soil fertility, humidity, density, genesis, and pollution levels. Soil microarthropod complexes can be used as indicators of soil conditions. Microarthropod studies in Hawaii were undertaken by Zimmerman (1948), Sprenger (1976), Mueller-Dombois et al. (1981), and Bellinger and Christiansen (1989), but numerical analyses of density and biomass were not performed.

In Hawaii Volcanoes National Park forests, feral pigs (*Sus scrofa*) are considered to be the most disruptive of the introduced species. Park managers and researchers have developed effective and efficient control measures. The study area is located in 'Ōla'a rainforest on the windward slope of Mauna Loa volcano, Hawaii Volcanoes National Park. Study sites include three units that were cleared of pigs 7, 4, and 2 yr ago, and one that had not been cleared (contained about 20 pigs km^{-2}). All study sites had the same type of soil and vegetation. Soil is volcanic; vegetation consists of open-canopy 'ōhi'a (*Metrosideros polymorpha*) and closed-canopy tree fern (mostly *Cibotium glaucum*). Microarthropods were driven out of 25-cm^3 soil samples through Berlese-Tullgren funnels, into alcohol.

The soil microarthropod fauna in 'Ōla'a forest consists mostly of Collembola (55% of the numbers, 45% of the biomass), Oribatida (19% of the numbers, 35% of the biomass), Acarida (14% of the numbers, 8% of the biomass), and Gamasida (6% of the numbers, 19% of the biomass). Protura, Prostigmatida, and Pauropoda show very low density and biomass. Soil Nematoda, Enchytraeina, Magascalecidae, Mollusca, and Insecta numbers and biomass did not differ between pig-free and infested areas. Over 65 % of the fauna is concentrated in the litter layer. Deep soil (15 to 30 cm) contains less than 10% of the

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numbers and 5% of the biomass. During forest recovery, surface soil density decreased from 1.65 to 0.55 kg cm^{-2}; density of microarthropods increased from 25,000 to 57,000 m^{-2}; and biomass increased from 240 to 800 mg m^{-2}. The group most sensitive to pig impact was Collembola; their species count doubled (7 to 16), density increased from 9600 to 32,200 m^{-2}, and biomass from 58 to 338 mg m^{-2}. Endemic Collembola increased during forest restoration from 2000 to 16,000 m^{-2}, and cosmopolitan species decreased from 2500 to 500 m^{-2}. Relationships between litter and soil life forms of Collembola changed during recovery succession. Litter-living animals increased 10-fold because litter density increased during forest recovery. Horizontal distributions of Collembola were examined in soil under 'ōhi'a and tree fern canopy: numbers were almost equal under both at all sites, but biomass under 'ōhi'a trees was much greater than under tree ferns.

The most significant impact of feral pigs on soil biota is trampling of the surface and compaction of upper soil horizons. The recovery process requires more than 7 yr. Analysis of soil microarthropods provides a valuable indicator of soil conditions and thus is an important indicator of the health of a forest ecosystem.

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Loss of Native Reptiles Associated with Introductions of Exotics in the Mariana Islands

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ABSTRACT: Few data are available for documenting long-term population trends of reptiles on Pacific islands. The best-studied populations are those on Guam that have received concentrated attention because of the introduction and subsequent irruption of the Brown Tree Snake, *Boiga irregularis*. In collaboration with Tom Fritts (U.S. Fish and Wildlife Service) and many others, I have tabulated the results of over 1000 person hours of visual censuses, 15,800 trap hours of adhesive trap sampling, and 10,000 trap nights of snake trap sampling covering 13 of the Mariana Islands. In the Marianas, dramatic declines have occurred since World War II in several species of native skinks and geckos. Some have been locally extirpated. Evidence that these disappearances were associated with introduced species was tested by comparing the present reptile community structure to that found on the island before arrival of the putative threat and by comparing the community composition on islands that now possess the putative predator/competitor/etc. with that on those islands where the introduced species is absent.

The gecko *Nactus pelagicus* has disappeared from Guam and Tinian, but is abundant in appropriate habitat on Rota, an island that differs from Guam and Tinian in not having experienced the introduction and irruption of

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the shrew *Suncus murinus*. The gecko *Perochirus ateles* has disappeared from the islands of Guam and Tinian. Its absence from Guam is associated with the introduction of the Brown Tree Snake, *Boiga irregularis*. The situation on Tinian is less well documented. The skink *Cryptoblepharus poecilopleurus* has disappeared from Guam. Although the evidence from the Marianas is insufficient to attribute this decline to any specific factor, the skink does not coexist with the Brown Tree Snake on any of the 69 islands (for which we have species lists) in their area of sympatry in Australasia. The probability of this occurring by chance is less than 0.005. *Emoia caeruleocauda* has declined dramatically on the three large islands of the Marianas (Guam, Tinian, Saipan), although it is still abundant in forested areas. It appears to have been displaced from more open habitat by the larger, very aggressive, kleptoparasitic, introduced skink *Carlia fusca*. Both local representatives of the genus *Gehyra* (*G. oceanica* and *G. mutilata*) have declined significantly on Guam, with the most likely agent being the introduced Brown Tree Snake.

In Australia, Papua New Guinea, the Solomon Islands, and in other Pacific areas, other native snakes and lizards (especially *Varanus*) have become noticeably rarer or have disappeared from informal censuses. The introduced toad *Bufo marinus* is thought to release allelo-chemicals that inhibit or stop the development of native amphibian species in syntopic waters in the Solomon Islands. Effective control measures for these introduced species are at present unknown; thus reptile conservation efforts must be directed at keeping the exotics from spreading to additional islands.

Impacts of Alien Species on the Avifauna of the Northwestern Hawaiian Islands

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ABSTRACT: Alien species have had a profound effect on the native avifauna of the Northwestern Hawaiian Islands (NWHI). Although somewhat limited in diversity, endemic bird species occur, or formerly occurred, on several of the NWHI. In addition to these native species, there is a wide variety of nonnative species that have been introduced through the actions of humans.

The history of Laysan Island perhaps best illustrates the power of even a single introduced species to alter an island's habitat and consequently alter the number and types of native birds found there. Rabbits (*Oryctolagus cuniculus*), which were introduced to Laysan Island about 1903, proliferated unchecked, denuding the island of vegetation and destabilizing the sand. Unknown varieties of insects, several endemic plant species, and three of five endemic land birds became extinct on Laysan Island during this period: the Laysan Rail (*Porzana palmeri*), the Laysan Millerbird (*Acrocephalus familiaris familiaris*), and the Laysan Honeycreeper (*Himatione sanguinea freethii*).

Although rabbits no longer occur in the NWHI, Roof Rats (*Rattus rattus*) and domestic mice (*Mus domesticus*) are present on Midway Atoll, and Polynesian Rats (*Rattus exulans*) are present on Kure Atoll. These rodents remain a constant threat to other islands. Rats arrived on Midway in 1943 and, by 1945, had caused the extinction of a translocated population of Laysan Rails, the last surviving population of this species. Rats have also been known to prey on seabird eggs, chicks, and even adults; ground- and burrow-nesting species are particularly vulnerable. Rats also feed on the apical and lateral buds of the native shrub *Scaevola sericea*, resulting in decreased plant vitality and growth.

Introduced plants pose additional threats to the avifauna of the NWHI. An introduced annual grass, sandbur (*Cenchrus ecinatus*), appears to be crowding out the native bunch grass (*Eragrostis variabilis*) on Laysan Island.

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Eragrostis variabilis is the primary nesting habitat for the endangered Laysan Finch (*Telespiza cantans*) (Morin and Conant 1990). Ironwood trees (*Casuarina* spp.) on Midway shade out ground cover and eliminate the native dune/shrub complex. Golden crown-beard (*Veresina enceloides*), found in disturbed areas on Midway and Kure Atolls, is believed to play a key role in the spread of avian pox (*Poxvirus avium*) by providing roosting sites for two nonnative avian disease vectors: a mosquito (*Culex quinquefasciatus*) and greenbottle flies (family Calliphoridae).

Exotic ants have recently been implicated in the deaths of nestling Laysan Finches and in the nest abandonment and hatching failure of several species of seabirds.

On Midway Atoll, canaries (*Serinus canaria*) and common mynahs (*Acridotheres tristis*), which were introduced in 1910 and 1974, respectively, have become established. These species compete for food with migratory shorebirds and could introduce, spread, or serve as reservoirs for avian diseases and parasites.

Managers and biologists continue to work toward solving the assortment of problems caused by alien species. There are currently plans to implement rat eradication measures at Midway Atoll. A 2-yr effort to eradicate sandbur (*C. ecinatus*) from French Frigate Shoals using herbicide treatments is showing promising results. Manual control of ironwoods at Midway Atoll is allowing native vegetation to return. Golden crown-beard (*V. enceloides*) is controlled by mowing, which limits the spread of avian pox. Ants in the NWHI will prove harder to control. The first step, a distributional survey of each island, has been completed and plans for control, at least near important nesting areas, are being formulated. At present, access to these islands is strictly controlled and techniques have been devised to minimize the chance for further introductions.

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When Endangered Species Are Aliens: Some Thoughts on the Conservation of Rare Species

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ABSTRACT: Recent attempts to conserve endangered and rare species have involved introducing such species to environments where they were not previously known to exist. These populations of endangered aliens become established in ecosystems free of the perturbations that caused population declines in the original habitat. Although this technique has been quite successful, particularly for birds, it is not without certain risks, some of which can be assessed or anticipated before the introduction, and some of which are quite unexpected. Endangered organisms in new habitats may behave like their alien counterparts elsewhere: changing the dynamics of the ecosystem, preying upon native, possibly rare organisms (e.g., Diamond 1990), and even causing changes detrimental to their own survival (see below). Like other aliens, these species may also begin to change, adapting to the novel environment. Depending on the size of the founder group, the rate of population growth, and other factors, genetic changes, notably genetic drift, may occur. These phenomena should be recognized in management programs for rare species, and factored in to management planning as well.

The endangered Laysan Finch (*Telespiza*

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cantans) is a case in point, and suggests caution for translocating rare species. Introduced to Pearl and Hermes Reef (PHR) in 1968, the finch flourished, establishing populations on each of four vegetated islands in the atoll. Comparative studies of morphology (Conant 1988), behavior (Morin and Conant 1990), and genetic variation (Fleischer et al. 1991) have shown the PHR finches to differ significantly in all respects from the parent finch population on Laysan, as well as indicating morphological and genetic differences among the four populations at PHR. It is not clear what caused the variation, although genetic drift and selection, which can act strongly on small populations, and environmental factors affecting development could account for the exceedingly rapid appearance of such dramatic variation. More ecological data are needed to assess impacts of the alien finch at PHR, but some changes are already obvious. A severe decline in the tussock grass (*Eragrostis variabilis*) appears inversely correlated with the increase of the finch populations there (Morin and Conant 1990). Finches feed on the seed of this grass and, on Laysan, members of the parent finch population choose it as a nest site more than 99% of the time (Morin and Conant 1990). Extirpation of the grass from PHR would mean the loss of a major component of the PHR plant community, although it appears finches will survive without it. Laysan Finches also feed on seabird eggs, but no formal studies of their impact on seabird reproduction at PHR has been undertaken.

Translocation of the Laysan Finch resulted in the successful establishment of four new populations of finches, although the sizes of these populations (> 50 to < 400) are small enough that they are at great risk of extinction from all the usual sources of uncertainty (i.e., demographic, genetic, environmental, catastrophic). Conservation biologists might well ask if it would have been better to safeguard the natural population of 10,000 finches on Laysan by vigilant monitoring and appropriate management than it was to create a metapopulation of endangered aliens that are now undergoing rapid genetic, morphological, and behavioral change as they irreversibly change the relatively pristine environment to which they were so carefully and expensively introduced.

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Strategies to Prevent Establishment of Feral Rabbits on Maui, Hawaii

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ABSTRACT: Although the Hawaiian Islands are well known to be highly vulnerable to biological invasion, the domestic European rabbit (*Oryctolagus cuniculus*) had come to be disregarded as a potential invader after more than a century without invasion on a major island. Haleakala National Park, on the island of Maui, initiated rabbit removal and monitoring in July 1990 after discovery of a reproducing rabbit population covering 25 ha in high-elevation (2070–2105 m) native shrubland. Because of the threat of feral rabbits to native biota, rabbit eradication was placed as

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the highest Park priority. The population is believed to have originated from as few as six unwanted cage-reared rabbits released by a pet owner in October 1989.

A total of 92 rabbits was removed from the 25-ha area of the infestation during August 1990–March 1991, through snaring, shooting, and live-trapping. Snaring, using a 75-cm length of steel cable with a dangling terminal noose, was the most effective control method in reducing the population, largely because rabbits followed discrete paths amongst the dense shrub/grass vegetation. Snaring became much less efficient as rabbit density became sparse and individual rabbits presumably became snare-shy. If snaring had not worked, there were no other obvious options for elimination of rabbits. Four additional rabbits were removed upslope of the initial infestation (elevation 2140–2500 m), at distances of 0.6–2.0 km from the main area of infestation, the last one on 7 May 1991. Monitoring of transects (assessment of rabbit pellet presence/absence, age, and abundance), combined with scouting and follow-up of reports of rabbit sightings by visitors and other Park employees, concurrently with control, allowed accurate assessment of numbers and location of remaining rabbits. Whether the four disjunct rabbit occurrences represent instances of wandering rabbits or separate releases is unknown.

Rabbits occupy small home ranges and are very likely to escape detection in the early stages of population buildup. At least initially, these domestic rabbits were more naive and vulnerable to a control program than their wild counterparts (lack of extensive burrowing, diurnal tendencies). They exhibited a spectacular potential for increase, however. Within a period of 1 yr (October 1989–October 1990), the founding population of six rabbits increased 15-fold. Without prompt attention and with a continued 15-fold rate of increase, there would have been 4.5 million rabbits on Maui by October 1994.

The small Indian mongoose, domestic or feral cats, and domestic or feral dogs are almost ubiquitous potential predators of rabbits on Maui. Whereas predator densities are very high at low elevation, densities are relatively low above 2000 m elevation and lower still on the northwest slope of Haleakala National Park because of an extensive predator trapping program instituted since 1980 to protect the endangered Nene (Hawaiian Goose) and Hawaiian Dark-rumped Petrel. Predation is probably a major factor limiting rabbit population establishment at lower elevations, but not at higher elevations, in Hawaii.

There is an abundance of pet rabbits on Maui. Many pet owners are irresponsible (and break the law), based on the dozens of reported instances of loose rabbits during 1990–1991. There is great potential for recurring outbreaks of rabbits on Maui in the future unless preventive steps are taken. The problem of preventing rabbit establishment does not seem to fall within the perceived jurisdiction or mandate of any State or County agency. The Park is still faced with the likelihood that release or escape of pet rabbits will pose a recurring problem.

Potential Effects of Alien Fruit Fly Eradication on Natural Areas of Hawai'i: An Exploratory Investigation in Haleakala National Park

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ABSTRACT: Although momentum is building for a complex Federal/State program to eradicate four pest fruit fly species (Diptera: Tephritidae) from the Hawaiian Islands, analyses of potential ecological effects on protected natural areas are almost nonexistent. Initial investigations on impacts on native biodiversity of a pest fruit fly monitoring

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and eradication program using male annihilation were conducted in Kīpahulu Valley of Haleakala National Park at four sites at 610–1525 m elevation in May–June 1990. The upper portion of the study area chosen, above 900 m, is one of the more pristine rainforest areas in Hawai'i, although nonnative strawberry guava (*Psidium cattleianum*) occurs sporadically as high as 1370 m. The lower portion of the study area is at the guava/native forest interface, believed to be the type of site with the most potential for negative effects of a fruit fly control program on nontarget organisms.

Although our sample size makes it impossible to draw definitive conclusions, we feel that preliminary assessment is warranted to establish hypotheses for future testing. Using standard chemical traps for three fruit fly species (one at each elevational site baited with trimedlure, one with methyl eugenol, and one with cuelure), left in the field for 4 weeks, these investigations (1) failed to detect Mediterranean fruit fly (*Ceratitis capitata*), (2) found melon fly (*Dacus cucurbitae*) present in very low numbers, (3) found that the Oriental fruit fly (*Dacus dorsalis*) is present as high as 1220 m and is locally abundant at the lower elevations, primarily in association with nonnative strawberry guava, and (4) trapped at least 23 native species of Drosophilidae, including local endemics. The abundance of nontarget native flies in methyl eugenol traps at the 610-m (648 specimens of 16 species in a single trap) and 915-m (151 specimens of 5 species) elevation trap sites suggests that some native flies may be attracted by dead Oriental fruit flies rather than by methyl eugenol. However, seven taxa of native flies were caught in the methyl eugenol trap at 1525 m in which no Oriental fruit flies were caught. Future investigations will use screening to exclude Oriental fruit flies from one set of methyl eugenol traps to determine which nontarget flies are attracted solely to methyl eugenol. The attraction of chemical lures for nontarget species such as these should be explored much more thoroughly. Much more detailed ecological studies throughout the year will be required before any serious evaluation of effects of an eradication program can take place. However, our preliminary data support the view that effects of eradication on native biological diversity pose a valid concern that must be adequately addressed through intensive research.